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Coastal Salt Marsh Systems in the U.S.: A Review of Anthropogenic Impacts

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ABSTRACT

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During the past century, human modification of environmental systems has greatly accelerated tidal salt marsh deterioration and shoreline retreat in many coastal regions worldwide. As a result, more than 50% of the original tidal salt marsh habitat in the U.S. has been lost. Numerous human activities have contributed directly or indirectly to wetland loss and alteration at local, regional, and global scales. Human impacts at the local scale include those that directly modify or destroy salt marsh habitat such as dredging, spoil dumping, grid ditching, canal cutting, leveeing, and salt hay farming. Indirect impacts, which can be even more significant, typically are those that interfere with normal tidal flooding of the marsh surface, alter wetlands drainage, and reduce mineral sediment inputs and marsh vertical accretion rates. These impacts usually develop over a greater period of time. At the regional scale, subsidence caused by subsurface withdrawal of groundwater, oil, and gas has submerged and eliminated hundreds of square kilometers of salt marsh habitat in the Chesapeake Bay, San Francisco Bay, and Gulf of Mexico. At the global scale, atmospheric warming due to increased burden of anthropogenic greenhouse gases and tropospheric sulfate aerosols appears to be strongly coupled to glacial melting, thermal expansion of ocean waters, and eustatic sea-level rise. Changes in coastal water levels ascribable to eustatic sea-level rise pose a long-term threat to the stability and viability of these critically important coastal systems.

ADDITIONAL INDEX WORDS: *Human activities, relative sea-level rise, subsidence, shoreline retreat, wetland loss.*

INTRODUCTION

Coastal salt marshes—areas vegetated by herbs, grasses, and low shrubs bordering saline water bodies—are unique ecosystems universally recognized for their exceptional ecological value (MITSCH and GOSSELINK, 1993; KENT, 1994; ALONGI, 1998; EISMA, 1998). They not only rank among the most highly productive ecosystems on earth but also provide critically important habitat for numerous aquatic and terrestrial organisms, many of which are of significant commercial and recreational value. In addition to being sanctuary refuges for many fish, wildlife, and waterfowl populations, these wetlands perform a number of vital chemical and physical functions, serving as sites of chemical contaminant retention and transformation, organic carbon production and export, groundwater recharge, sediment entrapment, shoreline erosion mitigation, and flood attenuation. Adjacent uplands are typically buffered by coastal wetlands against potentially catastrophic natural events (*e.g.,* hurricanes and tsunamis). Investigations of coastal wetlands must consider natural processes and human activities occurring in watershed areas as well as in adjacent estuarine and coastal marine waters because of the close coupling of these systems.

Salt marshes occur on intertidal shores in mid- and high latitude regions worldwide (Figure 1). They commonly flourish around high tide level in sheltered coastal embayments of the temperate zone where wave energy, tidal regime, and substrate conditions are favorable for their development. Within their geographical range, salt marshes are influenced by a array of local factors that control their development, most notably tidal characteristics (*e.g.,* tidal regularity and range), surface drainage, pedological conditions, sediment accretion (both organic and inorganic components), wave and current action, erosion, freshwater inflow, salinity, nutrient concentrations, shoreline structure, and marsh topography (CLARK and PATTERSON, 1985; FREY and BASAN, 1985; MITSCH and GOSSELINK, 1993; LEWIS, 1994; EISMA, 1998).

The relative rates of sediment accretion and coastal submergence determine the long-term stability of salt marsh systems (MITSCH and GOSSELINK, 1993). Accretion is defined as the net building of marsh soils via the deposition of mineral sediment and organic matter on marsh substrate (DELAUNE *et al.*, 1989; NYMAN *et al.*, 1993; BRYANT and CHABRECK, 1998). Submergence refers to all factors that raise the water level relative to the land surface including eustatic sea-level rise, subsidence of the land surface due to sediment compaction and crustal downwarping, and the ameliorating effects of sediment deposition (aggradation) (SASSER *et al.*, 1986).

Salt marsh systems are highly susceptible to submergence and erosion associated with rising sea level (CRAFT *et al.*, 1993). Present day salt marshes became established in many U.S. coastal areas during the past 3,000 to 4,000 years as the rate of relative sea-level rise during the late Holocene de-

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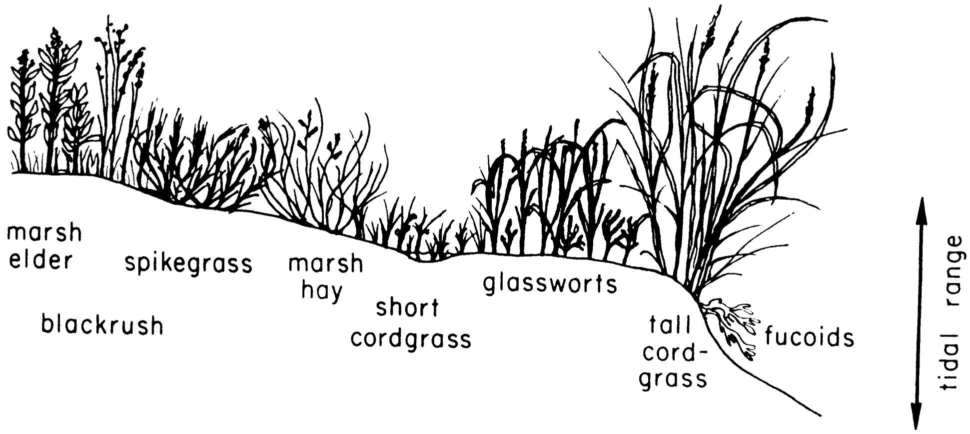

Figure 1. Diagrammatic section through a New England salt marsh showing a transect from the upland (left) to a tidal creek (right). Marsh plant species present from left to right in the diagram include: *Iva frutescens, Juncus gerardi, Distichlis spicata, Spartina patens, Salicornia europaea,* and *Spartina alterniflora. Fucus* and *Ascophyllum* represent the fucoids. Note tidal range on right. From VALIELA (1995).

clined from ~2.5 mm/yr to ~1 mm/yr (BRICKER-URSO *et al.*, 1989; WARREN and NIERING, 1993). Sea-level rise has continued to change during the past century as a consequence of atmospheric temperature changes (*i.e.,* volume expansion of warmed ocean waters and melting of polar ice sheets and continental glaciers) as well as tectonic and postglacial isostatic adjustments (deformation of ocean basins and subsidence or emergence of land) (VALIELA, 1995). Eustatic sea-level rise is estimated to be ~0.12–0.24 cm/yr (GORNITZ *et al.*, 1982; BARNETT, 1984; PELTIER and TUSHINGHAM, 1989). For the past several decades, the rate of relative sea-level rise has increased appreciably along much of the U.S. coast resulting in a net loss of coastal wetlands area in many regions (ORSON *et al.*, 1985; NYMAN *et al.*, 1993; DAVIS, 1997; WILLIAMS *et al.*, 1997). Coastal wetlands loss may be defined as the change from vegetated wetlands to: (1) submerged habitats; (2) nonvegetated wetlands (*e.g.,* mudflats); and (3) uplands or drained areas (BOESCH *et al.*, 1994).

ORSON *et al.* (1985) have shown that salt marshes respond to sea-level rise in the three ways: (1) they can retreat if the rates of coastal submergence exceed the vertical accretion of the marsh surface; (2) they can remain stable if sediment input equals the rate of coastal submergence so that surface elevations are maintained; or (3) they can expand both vertically and laterally if the rate of coastal submergence is less than the sediment accretion rate. The failure of salt marshes to keep pace with isostatic and eustatic sea-level rise is generally ascribed to insufficient sediment deposition on the marsh surface leading to accretion deficits (*i.e.,* vertical accretion less than relative sea-level rise) (BAUMANN *et al.*, 1984; STEVENSON *et al.,* 1986; WARD *et al.*, 1986; BRICKER-URSO *et al.*, 1989; NYMAN *et al.*, 1993; DOWNS *et al.*, 1994; EISMA, 1998). In deteriorating marshes, reduced sediment supply is commonly mediated by human activities, creating low values of marsh accretionary balance (*i.e.,* sediment accumulation rate minus relative sea-level rise rate) (WOOD *et al.*, 1989). These marshes are being replaced at a rather alarming rate by open water and shallow subaqueous flats (DELAUNE *et al.*, 1983; HATTON *et al.*, 1983; BAUMANN *et al.*, 1984; WHITE and TREMBLAY, 1995).

Human activities have been responsible for the alteration and loss of thousands of square kilometers of coastal salt marshes during the past century (KENNISH, 1992, 1997, 1999). This work provides a comprehensive examination of these anthropogenic impacts based largely on an analysis of published literature of studies conducted mainly along the Atlantic and Gulf Coasts of the U.S. Included in this effort is an analysis of small-scale impacts on local marsh habitats as well as large-scale impacts affecting extensive coastal wetland systems worldwide.

SCALE OF HUMAN IMPACTS

Human activities potentially threaten the viability of salt marsh systems on local, regional, and global scales. Direct impacts include those that result from the physical alteration and immediate loss of habitat during construction of bulkheads, dikes, weirs, levees, piers, docks, pipelines, revetments and other hard structures, as well as the excavation of canals, ditches, and oil drill sites (DEEGAN *et al.,* 1984; SASSER *et al.*, 1986; SWENSON and TURNER, 1987; DELAUNE *et al.*, 1989; TURNER, 1990; WHITE and MORTON, 1997; BRYANT and CHABRECK, 1998). Historically, the modification of coastal marshes for agricultural purposes (*e.g.,* draining and filling) and their reclamation for domestic and industrial development have substantially reduced viable wetlands habitat area during the past century (ADAM, 1990; ANDERSON *et al.*, 1992). Longer term, indirect impacts are also associated with some of these habitat disturbances. For example, the construction of impoundment dikes, water-control embankments, levees, dams for flood control, as well as canals and their associated spoil banks invariably alters the hydrology of these wetland systems, often interfering with normal tidal flooding and drainage, mollifying overland water flow, decreasing sediment supply to the marsh surface, and arresting vertical accretion. Natural stream diversions exacerbate the

Table 1. *Major causes and mechanisms of wetland loss in the northern Gulf of Mexico.*[a]

Cause	Primary Mechanism
Direct habitat change	Dredging, construction, filling in or over, erosion, prospecting machinery (marsh buggies)
Sea level rise	Increased flooding of plants
Subsidence increases	
Natural	Net loss in vertical accretion without compensation
Oil and gas withdrawal	Accelerated net loss in vertical
Soil drying	Soil shrinkage, net loss in vertical
Hydrologic Changes/Effects	
Saltwater balance	Physiological stress leading to plant community change or death
River levees	Restricted sediment supply
Sediment sources	Less sediment from less overbank flooding in rivers or marsh; delta switching
Canals	Change in sediment source and distribution, salinity and water levels; widening; channel theft
Spoil banks	Change in sediment source and distribution, salinity, and water levels; water movement over and under marsh
Hurricanes	marsh destruction
Boat wakes/waves	Bank erosion
Vegetation changes	
Quantity	Change in physiological responses to salinity, sediment trapping, organic deposition, flooding
Quality	Change in organic deposition, sediment trapping, intraspecific competition
Pollutants (brine, drilling fluids, and other)	Death of plants
Other	
Introduced pests	Death of plants by parasitic insect (primarily on alligator-weed)
Muskrat "eat outs"	Reduced vegetation cover leading to pond formation

[a] From Turner (1990)

problem through riverborne sediment deprivation (DELAUNE *et al.*, 1978). In a rapidly subsiding coastal zone, such as the Mississippi River Delta, accretion deficits are contributing to acute shoreline retreat and the conversion of marsh to open water along broad expanses of habitat. Table 1 lists various causes and mechanisms of coastal wetlands loss in the heavily impacted region of the northern Gulf of Mexico.

Local Impacts

Reclamation

Tidal salt marshes have been most directly and radically impacted by local land reclamation projects. The conversion of tidal salt marshes to flat coastal land for agricultural, residential, and industrial development has historically destroyed thousands of square kilometers of valuable wetlands habitat. The insidious, albeit irreversible loss of salt marsh habitat by land reclamation projects has been particularly acute in certain regions. For example, most tidal salt marshes originally surrounding San Francisco Bay have been drained and filled for agriculture, salt production, and other human needs. WATZIN and GOSSELINK (1992) have estimated that more than 50% of the original tidal salt marsh habitat in the U.S. has been eliminated by various human activities, most notably reclamation. The significance of this habitat reduction in terms of economic losses to recreational and commercial fisheries in the adjacent estuarine and coastal marine waters ranges into the multimillions of dollars. Aesthetic losses, in turn, are incalculable.

Grid Ditches

One of the most frequently used methods of mosquito control in coastal wetlands involves the digging of ditches at regular intervals on marsh surfaces to remove standing water. For instance, grid ditching has been used in more than 90% of New England salt marshes. Although this method of water management decreases the breeding habitat of mosquitoes in salt marshes, it physically alters extensive areas of the marsh surface. Grid ditching increases tidal flow to the upper marsh, and thus enables low marsh habitats to penetrate into high marsh areas. However, normal flow over the marsh during high tides is often impeded by man-made levees comprised of spoil banks built along the borders of the ditches. Thus, sediment delivery and natural accretion processes may be circumvented over much of the marsh (KENT, 1994). In addition to altering physical and hydrological characteristics of salt marshes, grid ditching appears to be detrimental to some marsh inhabitants, especially avifaunal populations (CLARKE *et al.*, 1984).

Because of the potential negative impacts of grid ditching, a new method of mosquito control in salt marshes was devised during the 1950s termed "Open Marsh Water Management" (OMWM). Consisting of a system of reservoirs and shallow canals constructed in the marsh surface, OMWM provides predatory fish (*e.g., Fundulus* sp.) ready access to mosquito breeding areas where they heavily graze on mosquito larvae. OMWM systems also may interfere with the hatching cycle of mosquito eggs. The net result is a significant reduction in the number of larval mosquitoes that survive to pupate as adults on the marsh surface (BUCHSBAUM, 1994). Investigations by HRUBY *et al.* (1985) confirm the effectiveness of this approach.

OMWM systems require less maintenance than grid ditches. They also have much less of an impact than grid ditches on the physical, hydrological, and biological characteristics of salt marshes. Unlike grid ditches which are connected to tidal creeks and which drain pannes and pools on the marsh surface, the OMWM reservoirs and canals are much more localized and therefore less damaging. The marsh is naturally flooded during spring high tides, and consequently sedimentation and accretion processes continue unabated.

Dikes and Impoundments

Marsh impoundment is often conducted for purposes of waterfowl and wildlife management, or for erosion and saltwater intrusion control. It usually involves the implementation of structural marsh management practices such as the use of levees, weirs, and flap gates. Adjustable water control structures are utilized for water level and salinity control in impoundment areas (CAHOON and GROAT, 1990; BOESCH *et al.*, 1994). These structures can contribute to significant floral and faunal changes.

Historically, dikes have been constructed to restrict tidal inundation so that salt marsh habitats can be manipulated by humans for agricultural and domestic purposes. For example, farmers have harvested extensive areas of salt marsh hay behind dikes; this vegetation provides fodder for livestock and mulch for gardens. Some salt marsh systems (*e.g.,* New England salt marshes) are still used for this purpose. Other salt marshes have been converted for animal grazing and the growth of specialized crops (*e.g.,* cranberries and rice). The drainage of coastal wetlands to create farmland largely destroys the character and function of these valuable habitats. Diking has played a major role in the conversion of salt marshes for agricultural land use.

Many hectares of wetlands habitat have been modified by impoundments or semi-impoundments, especially in the Louisiana coastal zone. An impoundment is defined as the complete hydrologic isolation of a management area, and a semi-impoundment, the partial hydrologic confinement of an area. Natural or artificial levees typically surround impoundments, hydrologically isolating them so that they are "disconnected" from regional riverine, tidal or estuarine systems (COWAN *et al.*, 1988).

Nearly 10% of Louisiana's coastal wetlands area consists of impoundments (TURNER, 1990), encompassing ~37,000 ha of state and federal lands (DAY *et al.*, 1990). The impoundment of salt marshes in Louisiana for water-level control to promote waterfowl utilization, as well as for agricultural development, has been a major factor in local wetlands habitat loss (BOESCH *et al.*, 1994). Marsh diking is an integral component of impoundment creation in coastal wetland and upland regions. Marsh habitats behind constructed dikes may become markedly degraded over time due to reduced salinities associated with diminished tidal flushing. Altered flow in tidal creeks behind the dikes frequently impacts water quality. A pervasive and persistent change of marsh floral and faunal communities also commonly develops behind the dikes. For instance, reduced salinities may favor the invasion of brackish water plant species such as the common reed (*Phragmites australis*) and cattails (*Typha* spp.), with an accompanying replacement of many species of marsh fauna (ROMAN *et al.*, 1984; BEARE and ZEDLER, 1987). In addition, marsh sedimentation rates often vary appreciably with increases or decreases in organic matter production on the marsh surface (ZEDLER *et al.*, 1980). Impoundments behind marsh dikes frequently lower inorganic sediment inputs from land drainage, causing reduced sediment accretion rates in the altered marsh habitat. Despite the decreased sediment accretion rates, dikes strategically located along the upland edge of marshes may actually retard or temporarily preclude the normal migration of salt marshes in the face of rising sea levels.

BRYANT and CHABRECK (1998) investigated the vertical accretion of impounded tidal salt marsh at four sites in the Chenier Plain of southwestern Louisiana. Results of their work indicate that impoundment levees apparently preclude flood delivery of mineral sediment during storms, thereby ameliorating accretion of the impounded marshes. This is reflected in the lower marsh surface elevation in the impoundment marsh habitat, than in natural marshes, that arises from hydrologic isolation from tidal sediment subsidies, storm effects, and substrate oxidation during forced drying. In 1994, the elevation of impounded marshes was 20–30 cm lower than that of natural marshes in the study area. The levee system clearly forms a barrier to water and suspended sediment movement onto the impounded marsh surface. The reduced accretion rates that culminate appear to be insufficient for the impounded marshes to maintain their location in the intertidal zone.

Dredging, Leveeing, and Canal Cutting

Canal dredging is a common practice in tidal salt marshes, especially in many areas of the Gulf of Mexico such as southern Louisiana (BOESCH *et al.*, 1994), where operations of the oil and gas industry grew exponentially during the 20th Century. The Louisiana coastal zone is a dynamic system that has been built by a series of six major deltaic episodes over the past 7,000 years (Figure 2) (TURNER and RAO, 1990). According to ROBERTS (1997), the six major delta complexes collectively cover an area of ~30,000 km^2. While wetland loss has been a natural part of the 1,000 cycle of delta lobe development and decay off Louisiana, human activities hastened habitat losses at an unprecedented rate after World War II. For example, numerous canals and their associated spoil banks in northern Gulf of Mexico tidal salt marshes have been constructed to service the oil and gas industry, most notably for the transport of drilling equipment, installation of pipelines, excavation of drilling sites, and recovery of oil and gas (TURNER, 1990). In addition, other artificial channels have been constructed as navigational waterways. TURNER (1990) reports that all of the canals in the Louisiana-Mississippi coastal zone cover 2.3% of the coastal wetlands area.

The dredging of oil and gas canals places significant pressure on Louisiana's wetlands by altering salinity, water levels, as well as the input and dispersal of sediments across the marsh surface. The canals and associated spoil banks, weirs, plugs, and levees modify hydrologic processes and may greatly reduce the mineral sediment supply necessary for vertical accretion of the marsh surface. Spoil banks, in particular, lessen overbank flooding and sediment input to the marshes. A 50% reduction in suspended sediment load in the Mississippi River during the past 50 years (due to changes in land

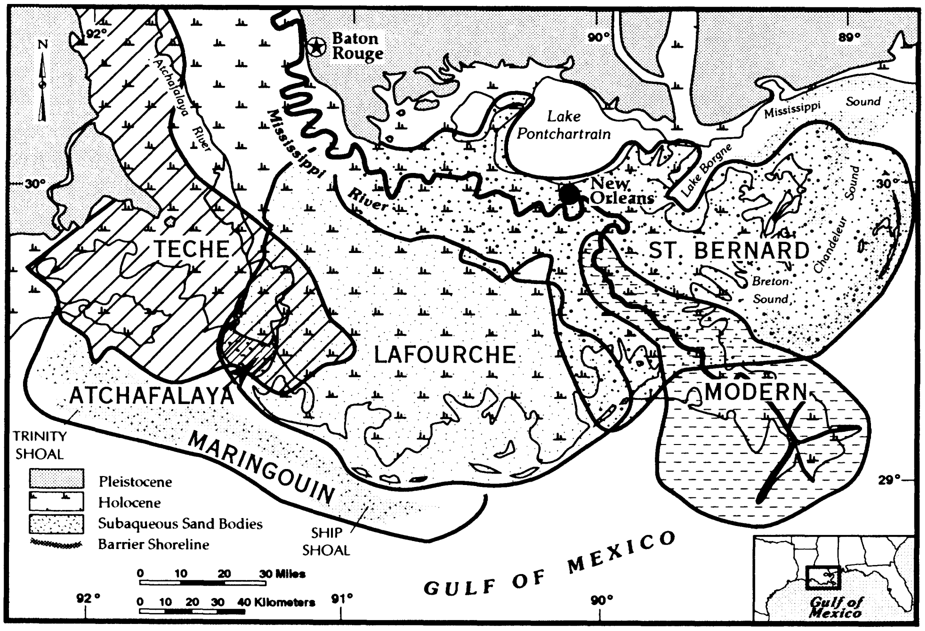

Figure 2. Mississippi River Deltaic Plain showing six distinct delta lobes deposited during the past 7,000 years. From PENLAND *et al.* (1988) and BOESCH *et al.* (1994).

use in watershed areas) has further decreased sediment delivery to the marshes (TURNER and RAO, 1990). In addition, the spoil banks and streamside levees raise land elevation and cover marsh plants, thereby accelerating marsh habitat loss and yielding unsuitable conditions for wetlands vegetation (DEEGAN *et al.*, 1984). The altered natural hydrology associated with the extensive levee system also impacts soil chemistry, which can hinder plant growth. SWENSON and TURNER (1987), investigating partially-impounded wetlands habitat in southern Louisiana, observed that spoil banks profoundly affect the marsh water-level regime and the biotic and physical factors important in land-building processes in an otherwise sinking deltaic coast. CRAIG *et al.* (1979) estimate that, for every hectare of canal dredged on a salt marsh, 2.5 ha of spoil banks are created in coastal Louisiana.

Diminished salt marsh accretion associated with canal and spoil bank construction is exacerbated by dredge and fill activities which directly convert wetlands habitat to open water or upland habitats. Dredging within or adjacent to tidal salt marshes also increases turbidity, which commonly leads to secondary impacts on biota. Wetland filling and the construction of artificial levees have reduced the original 2,000 km^2 area of San Francisco Bay salt marshes to only 85 km^2 (NICHOLS *et al.*, 1986), and they have contributed to large losses of tidal wetlands in the Mississippi Delta region (STEVENSON *et al.*, 1988). As shown by SASSER *et al.* (1986), the rate of salt marsh loss in western Barataria Bay, Louisiana, escalated rapidly during the 1945 to 1980 period (Table 2).

Of the 37,000 km^2 of coastal wetlands in the United States, 78% are associated with estuaries and barrier islands along the Gulf and South Atlantic coasts (GOSSELINK and BAUMANN, 1980; STEVENSON *et al.*, 1988). The wetlands in southern Louisiana alone constitute ~41% of the U.S. coastal wetlands (SWENSON and TURNER, 1987). DEEGAN *et al.* (1983, 1984) have attributed 25% to 39% of the tidal salt marsh loss in the Mississippi River Delta region of southern Louisiana to canal and spoil bank construction. Here, the rate of shoreline retreat exceeds 100 km^2/yr (GAGLIANO *et al.*, 1981). SCAIFE *et al.* (1983) estimated that the annual wetland loss rate in southern Louisiana is ~0.8% of the existing marsh. Freshwater marshes subjected to saltwater intrusion exhibit the highest rates of marsh loss (SASSER *et al.*, 1986). In some areas (*e.g.,* New Orleans), levees constructed for flood protection have hastened the rate of wetland loss by halting the landward migration of the retreating marshes (SALINAS *et al.*, 1986). However, much of the salt marsh inland migration in the Gulf of Mexico appears to be closely coupled to increasing marine intrusion and flooding associated with the natural cycle of delta building and decay (NYMAN *et al.*, 1995).

Using aerial imagery, TURNER and RAO (1990) investigated the patterns of coastal wetland loss in the Louisiana coastal zone. They identified five types of wetland changes over a 22-year study period: (1) spoil bank-parallel pond formation; (2) pond formation with apparent random distribution for the smallest ponds and clumped distribution for the largest ponds; (3) semi- or complete impoundment and resulting open water formation; (4) cutting off of stream channels upstream of where a spoil bank crosses a natural channel; and (5) erosion at the land-water interface. Their work demonstrated that the conversion of wetlands to open water involves a disintegrating process whereby interior salt marsh habitat fragments into small and then larger open water bodies. During this process, habitat breakup in the wetland interior predominates over shoreline erosion. TURNER and RAO (1990) also

Table 2. *Marsh area in 1945, 1956, 1969, and 1980, and rates of change during 1945–56, 1956–69, and 1969–80 in Barataria Basin marshes.*[a]

	March Area By Year				Rate of Change		
Category	1945 km^2	1956 km^2	1969 km^2	1980 km^2	1945–56 km^2/yr	1956–69 km^2/yr	1969–80 km^2/yr
Marsh (<10% water)	528.33	472.86	246.35	119.16	−5.04	−17.42	−11.56
Marsh (10–25% water)	45.84	73.92	179.52	147.03	1.24	8.12	−2.95
Marsh (25–40% water)	10.75	23.03	37.16	76.38	1.11	0.12	3.57
Marsh (40–60% water)	1.14	2.54	24.16	42.74	0.13	1.66	1.69
Marsh (60–80% water)	2.65	3.41	1.53	27.45	0.07	−0.14	2.36
Water	280.98	294.21	322.89	382.70	1.20	2.21	5.44
Developed	19.27	20.80	39.41	71.69	0.13	1.43	2.93
Natural levee	52.21	40.92	43.99	31.29	−1.04	0.24	−1.15
Canal[b]	3.94	13.40	48.94	42.79	0.85	2.73	−0.56
Other[c]	7.36	6.34	9.25	9.01[d]	—[d]	—[d]	0.11[e]

[a] From Sasser *et al.* (1986).
[b] Because canals and their associated spoil deposits are narrow, linear features, these numbers contain an undetermined error.
[c] This category includes Bayou Lafourche, the beach, and Louisiana Highway 1. Because these are narrow, linear features, the numbers may not closely approximate the actual areas of the features and are included only as an index of relative value.
[d] The 1980 datum for Louisiana Highway 1 was not generated and is not included in this entry.
[e] This entry represents only Bayou Lafourche since the beach and Louisiana Highway 1 represent <0.01% of the study area.

determined that canals and their spoil banks are spatially related to wetland-to-water conversions. Hydrological impacts of the canals and spoil banks affect the conversions up to 2–3 km away from the canals, and thus are a major causative factor of wetland loss in the system.

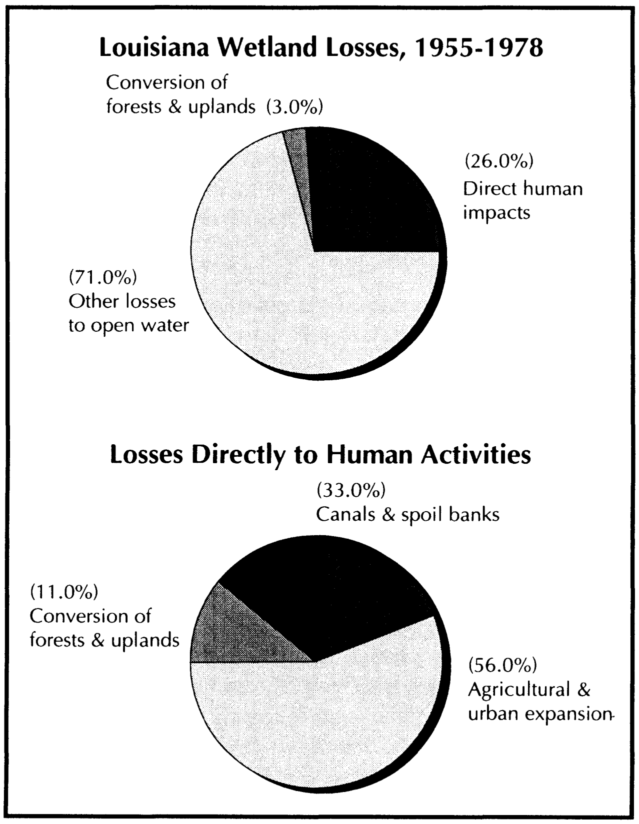

Figure 3. Causes of wetland losses in the Louisiana coastal zone, 1955 to 1978. From PENLAND *et al.* (1988) and BOESCH *et al.* (1994).

SALINAS *et al.* (1986) advocated the following management strategies for minimizing canal-induced tidal salt marsh loss: (1) prohibit unnecessary canal building; (2) backfill canals with previously stripped spoils; (3) design canals to flow down naturally occurring channels; (4) distribute the canal spoils evenly instead of building spoil banks; and (5) barricade crossing of canals at natural streams so that they do not divert waters from them. SWENSON and TURNER (1987), in turn, proposed the following options to minimize the adverse impacts of canal spoil banks on marsh hydrology: (1) backfill the canals; (2) place gaps in the spoil banks, (3) use weirs on flap gates; and (4) employ alternative dredging techniques that do not require spoil bank construction.

The indirect effects of canals and navigation channels may overshadow direct wetland destruction activities. For example, BOESCH *et al.* (1994) disclosed that direct human impacts on coastal wetlands in Louisiana—draining, dredging, spoil deposition, and subsequent widening of canals and navigation channels—only accounted for ~26% of the total wetland loss between 1955 and 1978, with canals and spoil banks being responsible for one-third of these losses (Figure 3). Indirect effects of canals and navigation channels—hydrologic modification, interference with floodwater flow, saline water intrusion into freshwater areas, and altered wetland drainage—were much more significant, causing most of the total wetland losses in this area during the 1950s, 1960s, and 1970s.

Altered flooding and salinity regimes associated with the construction of artificial levees have also contributed to the deterioration of wetland forests in coastal Louisiana (CONNER and BRODY, 1989). Many coastal forests in Barataria Basin and Verret Basin no longer receive sufficient amounts of sediment-laden waters to build adequate surface elevation to offset the effects of ongoing subsidence and sea-level rise. For example, construction of the Atchafalaya protection levees cut off sediment inputs and led to severe flooding problems in bottomland hardwood forests of the Verret Basin primarily due to increased subsidence and secondarily to upland runoff, backwater flooding, and rainfall. The bottomland hardwood species are less tolerant of flooding conditions than cypress-tupelo swamp vegetation, and hence are gradually being replaced by the swamp species. In upper Barataria Basin, much of the hardwood tree land cover has been cleared for development, and levees are now used for agricultural, industrial, and residential purposes. Hardwood forests in the Barataria and Verret watersheds, therefore, have declined significantly as a result of both direct and indirect impacts of human activities.

In summary, most anthropogenic impacts in the Louisiana coastal zone have been associated with the development of an extensive network of canals dredged for navigation, drainage, oil well access, and pipelines. Canal construction has modified hydrologic patterns and degraded the marsh interior. Flood-control levees have substantially reduced the sediment supply to most of the interdistributary marshes (SASSER *et al.*, 1986). Diminished vertical accretion has resulted in the conversion of much coastal salt marsh habitat to open water systems.

Dams, Reservoirs, and Land-use Changes

Sediment loads to the lower Mississippi River decreased after 1930 in response to land-use changes in watersheds and the implementation of various initiatives to control soil erosion. Reforestation programs and the abandonment of agricultural lands substantially reduced sediment yields from watersheds (STEVENSON *et al.*, 1988). In addition, the construction of dams and reservoirs on major tributaries (*e.g.,* the Arkansas and Missouri Rivers) caused a sharp drop in sediment loadings to the lower Mississippi River in the 1950s (MEADE and PARKER, 1984). Suspended sediments transported downstream are now slowly settling in basins behind the dams and in the reservoirs, although these man-made depositional sites have not yet filled (KESEL, 1987; TURNER, 1990). Annual suspended sediment loads in the lower Mississippi River declined from $\sim 380 \times 10^6$ t in the early 1900s to less than $\sim 240 \times 10^6$ t in the 1950s and early 1960s (STEVENSON *et al.*, 1988). During the 20-year interval from 1963 to 1982 suspended sediment loads in the lower Mississippi River decreased by $\sim 50\%$ (KESEL, 1988, 1989).

Artificial levee systems have virtually eliminated sediment influx to marsh surfaces in many areas of the Mississippi River Delta, resulting in accretion deficits of 4.1 to 8.1 mm/yr (DAY and TEMPLET, 1989), which equals an inorganic mineral deficit of 400 to 2,500 g/m^2/yr (BOESCH *et al.*, 1994). Sediments diverted downstream are shunted across the Mississippi Delta region and debouched into deeper waters at the edge of the continental shelf (STEVENSON *et al.*, 1988). Other sediments originating from runoff of upland areas collect in a system of drainage canals that flow directly to estuaries and embayments and bypass marsh habitats (BOESCH *et al.*, 1994).

Regional Impacts

The relative elevation of the sea and coastal salt marshes is dynamic, constantly responding to variable global factors (*i.e.,* changing volume of the ocean basins and total amount of ocean water linked to tectonic processes and glacial effects, respectively) and regional factors (*i.e.,* subsidence caused by fluid withdrawal and sediment compaction) (DELAUNE *et al.*, 1987a). Problems exist in attempting to separate eustatic from other components of relative sea-level change (*i.e.,* tectonic, isostatic, geoidal, and sedimentation rates) (THOM and ROY, 1985). The rate of change of relative sea-level rise differs considerably along the U.S. coastal shoreline. Along the Atlantic Coast, PATRICK and DELAUNE (1990) emphasize that the change in water level is mainly ascribable to sea-level rise, although some areas exhibit higher rise due to changes in surface water hydrology or subsurface water extraction.

The marsh surface is subsiding more rapidly along the Gulf Coast because of the compaction of sediments and downwarping of underlying crust (*e.g.,* Mississippi River Delta Plain, Barataria Basin, and Atchafalaya Basin), modification of surface water hydrology, and withdrawal of subsurface oil and gas (*e.g.,* southeast Texas Coast). The highest subsidence rates occur within the influence of the Mississippi River Delta at the geosyncline depocenter where subsurface sediment compaction and geosynclinal downwarping are greatest (PENLAND and RAMSEY, 1990; TURNER, 1990; CALLAWAY *et al.*, 1997). Here, the sedimentation rates must be sufficient to compensate for considerable subsidence to enable the marsh to maintain its elevation within the tidal range (SASSER *et al.*, 1986; DELAUNE *et al.*, 1989). Along the Louisiana Coast, the increase in water level due to subsidence is 3 to 5 times greater than eustatic changes (DELAUNE *et al.*, 1989). Over a 40-year period (1947 to 1988), the relative rate of sea-level rise along the Louisiana Coast exceeded 1 cm/yr in many areas (Figure 4). The resultant elevated submergence rates (Table 3) have degraded salt marsh habitats.

The Pacific Coast is not experiencing the general subsiding conditions which characterize the Gulf Coast. However, there are some local areas of higher subsidence resulting from subsurface water extraction. On a regional basis, coastal marshes along the Pacific Coast generally are keeping pace with the small increase in global sea-level rise and thereby maintaining their elevation above sea level (PATRICK and DELAUNE, 1990; THOM, 1992).

Regional differences in accretion rates are also evident in U.S. coastal marshes, although in many marsh systems the sediment accretion rates are keeping pace with sea-level rise (Tables 4, 5) (CALLAWAY *et al.*, 1997). Exceptions exist in the Mississippi Delta region and other Gulf coastal areas where salt marsh accretion rates resulting from accumulation of mineral sediment and organic matter have generally not been rapid enough to balance sea-level rise (Table 4, Figure 5) (DELAUNE *et al.*, 1978, 1983; DELAUNE *et al.*, 1987b; HATTON *et al.*, 1983; REED, 1989; PENLAND and RAMSEY, 1990; BOESCH *et al.*, 1994; CALLAWAY *et al.*, 1997; WHITE and MORTON, 1997). In addition, erosion is a serious problem, especially along the Mississippi and Louisiana shorelines and parts of the southeast coast of Texas (Figure 6). Along the Pacific Coast, (LETZSCH and FREY, 1980; BENNINGER and CHANTON, 1985; SHARMA *et al.*, 1987b), Mid-Atlantic Coast (KEARNEY and WARD, 1986; OERTEL *et al.*, 1989; KEARNEY and STEVENSON, 1991; CRAFT *et al.*, 1993; KEARNEY *et al.*, 1994), and New England Coast (BRICKER-URSO *et al.*, 1989; ANDERSON *et al.*, 1992;

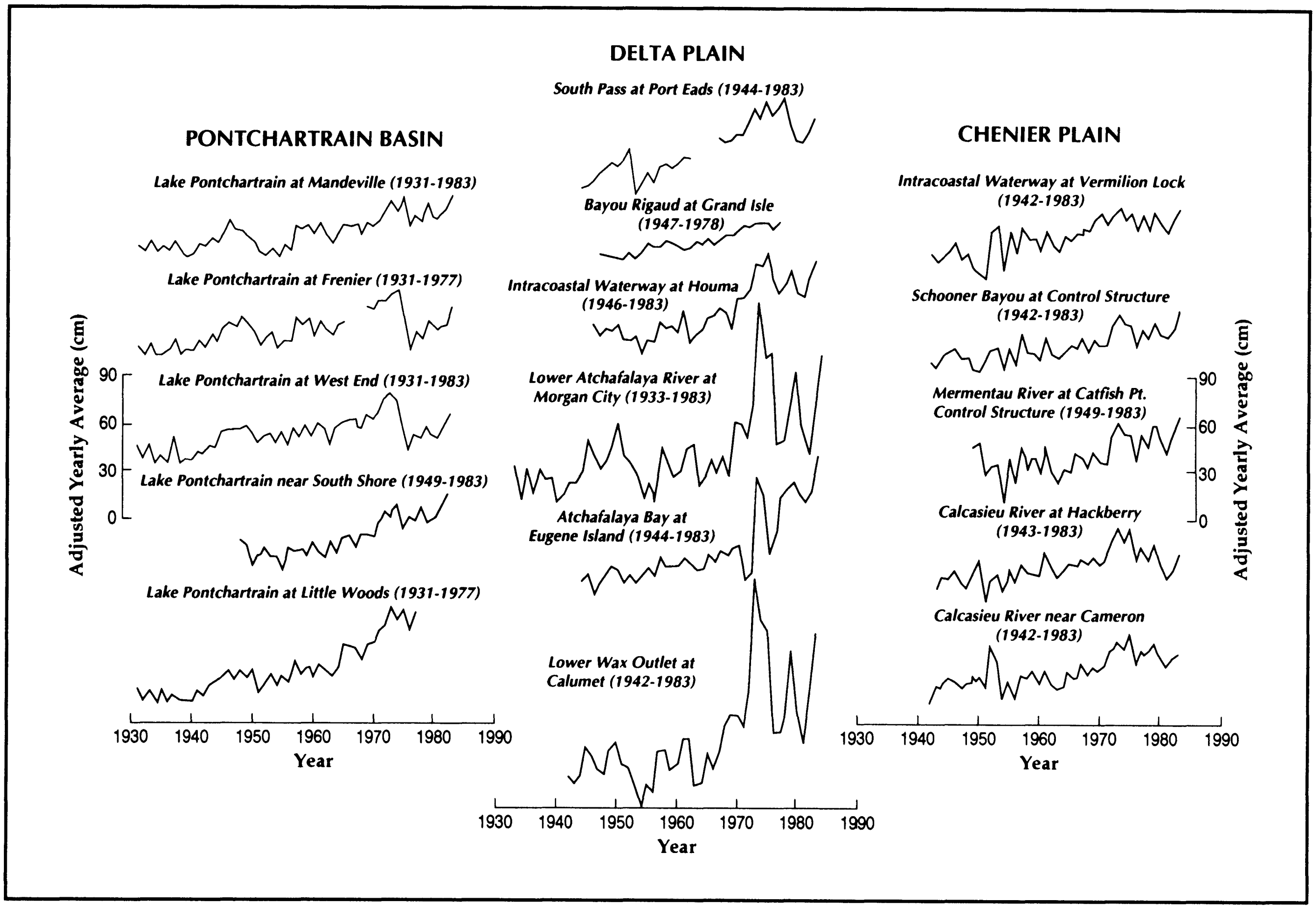

Figure 4. Tide gauge records illustrating that relative sea level has risen at an average of ~1 cm/yr at various locations along the Louisiana Coast, although there have been periods of greater sea-level rise as well as periods of relatively stable conditions. From PENLAND and RAMSEY (1990) and BOESCH *et al.* (1994).

WARREN and NIERING, 1993), vertical accretion typically has been sufficient to compensate for sea-level rise and subsidence to maintain viable salt marsh systems. Accretion rates of salt marshes are affected by several key factors such as plant community biomass, mineral sediment deposition, sediment organic matter accumulation rates, proximity of sediment sources and creek systems, relative elevation, tidal range, and local rates of subsidence (LETZSCH and FREY, 1980; PETHICK, 1981; STUMPF, 1983; STEVENSON *et al.*, 1986; STODDART *et al.*, 1989; BOESCH *et al.*, 1994; NYMAN *et al.*, 1995; CALLAWAY *et al.*, 1997).

Table 3. *Submergence rates in the Lake Barre/Madison Bay area of Terrebonne Parish, Louisiana, estimated from tide gauge analysis.*[a]

Station	Period of Record	Submergence Rate (cm/yr)
Houma, Intracoastal Waterway	1962–1982	1.94 ± 0.39
Cocodrie, Bayou Petit Caillou	1969–1983	0.60 ± 0.32
Valentine, Bayou Lafourche	1966–1982	1.62 ± 0.29
Bayou Blue, near Catfish Lake	1976–1983	1.02 ± 0.46
Golden Meadow, Bayou Lafourche	1959–1979	2.33 ± 0.27
Leeville, Bayou Lafourche	1957–1983	0.74 ± 0.12
Mean Values	1964–1982	1.38 (SD = 0.70)

[a] From Penland *et al.* (1988).

Salt marsh deterioration can proceed rapidly when vertical accretion is arrested and eustatic sea-level rise and/or subsidence continue unabated. Ongoing sedimentation—both mineral sediment and peat production—is necessary to maintain the salt marsh surface within the intertidal zone and offset subsidence effects. However, episodic sedimentation events associated with hurricanes and other major storms are responsible for much of the sediment accumulation on some salt marsh systems. These events may be sufficient to negate the vertical accretion deficit in some areas (NYMAN *et al.*, 1995). Despite the great importance of vertical accretion in maintaining viable salt marsh systems, much information is lacking on accretionary processes that operate in the U.S. coastal zone (KEARNEY *et al.*, 1994; PARKER, 1994).

As inferred above, coastal wetland loss in the U.S. derives from interactive processes involving both natural and anthropogenic factors. The relative significance of these fundamentally different groups of factors varies considerably along

Table 4. *Rates of relative sea-level rise and vertical accretion in Atlantic and Gulf Coast marsh systems.*[a]

Site	Marsh Type	Relative Sea-Level Rise (cm/yr)	Accretion Rate cm/yr (20)
Barnstable (MA)	S.a. (1)	0.23	0.15–0.27
	S.a. (1)		0.34–0.79
	S.a. (1)		1.8
Narragansett Bay	S.p. (2)	0.26	0.24
	S.a. (2)		0.25–0.60
Delaware Bay	S.p. (3)	0.30	0.44–0.59
	S.p. (4)		0.47
	S.a. (5)		0.42–0.78
	S.a. (6)		0.32–0.45
	S.a. (6)		0.26–0.43
	S.a. (6)		0.4
Long Island Sound	S.p. (7)	0.22	0.35
	S.p. (8)		0.20–0.66
	S.p., S.a. (9)		0.54–0.81
	S.a. (10)		0.47–0.63
	S.a. (11)		0.20–0.43
	S.a. (12)		0.25–0.47
Chesapeake Bay	N.A. (13)	0.35	0.18–0.74
Georgia	S.a. (14)	0.27	0.26–1.5
South Carolina	S.a. (15)	0.34	0.14–0.45
	S.a. (15)		0.13–0.25
Louisiana	S.a. (16)	0.92	0.59–1.4
	S.a. (17)		1.35
	S.a. (18)		0.75–1.35
	N.A. (19)		0.81–1.4

[a] From Bricker-Urso *et al.* (1989).
S.a. is *Spartina alterniflora*.
S.p. is *Spartina patens*.
N.A. means type of marsh not identified.
(1) Redfield (1972), (2) Bricker-Urso *et al.* (1989), (3) Drier (1982), (4) Church *et al.* (1981), (5) Chrzastowski *et al.* (1987), (6) Sharma *et al.* (1987a), (7) McCaffrey and Thompson (1980), (8) Harrison and Bloom (1977), (9) Siccama and Porter (1972), (10) Armentano and Woodwell (1975), (11) Richard (1978), (12) Flessa *et al.* (1977), (13) Ward *et al.* (1986), (14) Goldberg *et al.* (1979), (15) Sharma *et al.* (1987b), (16) Hatton *et al.* (1983), (17) Delaune *et al.* (1981), (18) Delaune *et al.* (1978), (19) Delaune *et al.* (1987b), (20) Hicks *et al.* (1983).

Table 5. *Relative sea-level rise and salt marsh accretion rates at sites in the Pacific Northwest (Oregon and Washington).*[a]

Site	Annual Relative Sea-Level Rise (mm)	Sampling Stations	Accretion Rate (mm/yr)
Nisqually Delta	2.0	Mid-marsh	2.3
		Progading edge	3.0
		Dike depression	nd[b]
		Eroding edge	3.0
Padilla Bay	0.8	Mid-marsh	4.5
Washington Harbor	0.6	Mid-marsh	3.0
Grays Harbor (Elk River)	1.3	Mid-marsh diked	1.6
Salmon River	1.7	Mid-marsh open	3.0

[a] From Thom (1992).
[b] nd = no data.

coastal regions nationwide. For example, the effects of human-induced coastal subsidence associated with groundwater, oil, and gas withdrawal may greatly exceed the effects of subsurface sediment compaction and eustatic sea level changes at some locations (*e.g.,* the Galveston-Houston area on the upper Texas Coast) (TURNER and CAHOON, 1987; WHITE and TREMBLAY, 1995; WHITE and MORTON, 1997). The rates of compactional subsidence and eustatic sea-level rise along the upper Texas Coast may range up to 13 mm/yr, whereas the rates of human-induced subsidence may approach 120 mm/yr (WHITE and TREMBLAY, 1995). In other areas (*e.g.,* the northern Gulf Coast), vertical accretionary deficits on salt marsh surfaces caused by sediment deprivation associated with anthropogenic alteration of environmental systems (*e.g.,* construction of stream diversions, water control embankments, levees, dams, and impoundments) and natural factors (*e.g.,* delta abandonment, compactional subsidence, geosynclinal downwarping, long-term climate change, and severe storms) are leading to the rapid retreat of coastal marshes (DELAUNE *et al.*, 1978; SWENSON and TURNER, 1987; TURNER and RAO, 1990; WHITE and TREMBLAY, 1995; BRYANT and CHABRECK, 1998).

Along the Louisiana and Mississippi coastlines, where wetland loss rates are accelerating due to the local effects of subsurface fluid withdrawal, rapid submergence, erosion, and elimination of emergent vegetation (BAUMANN *et al.*, 1984; SASSER *et al.*, 1986; REJMANEK *et al.*, 1988), hydrologic modifications of salt marshes are extensive, and elements of climate and the rate of sediment supply have changed. In the Mississippi Delta Plain, for example, there is a deficiency of mineral sediments to support vegetative growth and marsh surface elevation. Thus, submergence, increased flooding, and accompanying saltwater intrusion stress plant communities (by degrading plants not saltwater tolerant), reduce plant productivity, and decrease marsh surface elevation, thereby promoting wetland loss (DELAUNE *et al.*, 1994). In addition, salt-tolerant plants may not replace freshwater spe-

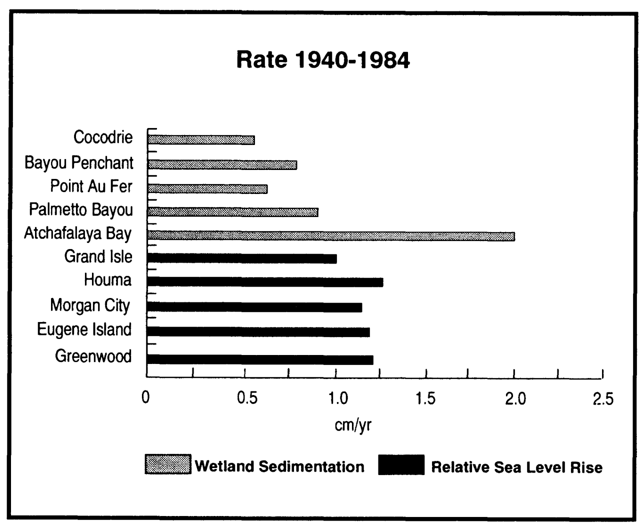

Figure 5. Rates of wetland sedimentation and relative sea-level rise at various coastal sites in the northern Gulf of Mexico. Note that sedimentation rates have failed to keep pace with relative sea-level rise except in areas of active deposition (*e.g.,* near the Atchafalaya Delta). From PENLAND *et al.* (1988) and BOESCH *et al.* (1994).

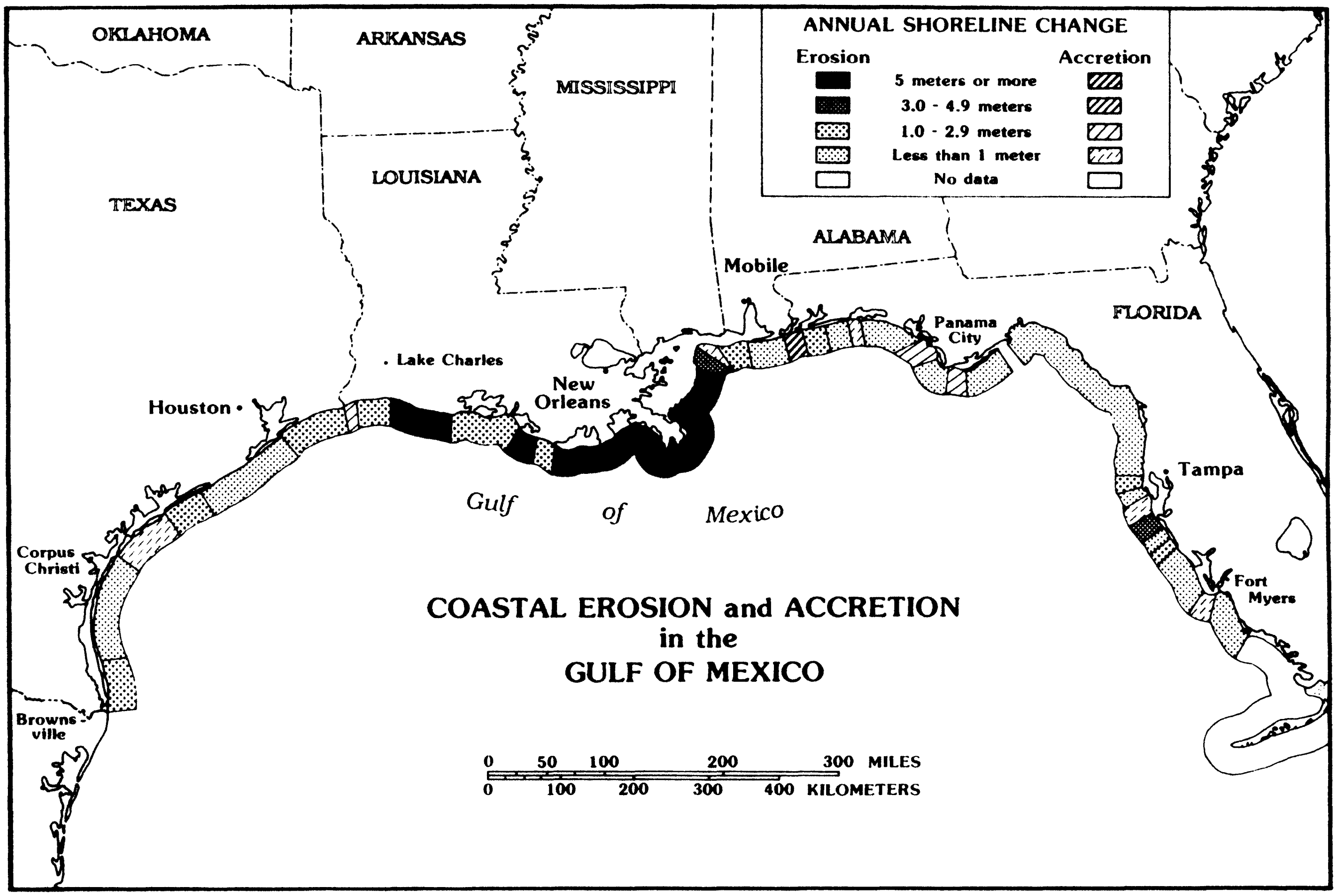

Figure 6. Map of the Gulf of Mexico displaying the general pattern and values of erosion and accretion in the coastal zone. Rates are in m/yr. From DOLAN *et al.* (1985) and DAVIS (1997).

cies rapidly enough to protect the marsh substrate from erosion (SASSER *et al.*, 1986). Diminished accumulation of organic matter arising from deficient plant production largely accounts for much of the observed vertical accretion deficits (NYMAN *et al.*, 1993). Vertical marsh accretion rates along the Louisiana Coast are ~0.6 to 0.8 cm/yr, which cannot maintain the marsh elevation because the submergence rates range from 1 to 3 cm/yr (DELAUNE *et al.*, 1989). Wetland losses along the Louisiana coastline alone between 1955 and 1978 ascribable to both direct and indirect human impacts ranged from 30% to 59% (BOESCH *et al.*, 1994). The most acute losses of habitat were incurred during the period from the 1940s through the 1960s (Figure 7).

Subsurface Fluid Withdrawal

In various regions of the U.S., subsurface fluid withdrawal is the primary cause of coastal wetland submergence and the loss of emergent vegetation. For example, increasing groundwater withdrawal in lower Chesapeake Bay has led to major subsidence in the coastal zone. At Portsmouth, the total subsidence between 1918 and 1990 amounted to nearly 22 cm (DAVIS, 1987). Similar findings have been reported for population centers along the southeast Atlantic Coast (*e.g.,* Savannah, Georgia) (KEARNEY and STEVENSON, 1991) and the upper Texas Coast (WHITE and TREMBLAY, 1995). Elevated subsurface water extraction also has created local conditions of higher subsidence along the West Coast, such as in South San Francisco Bay. PATRICK and DELAUNE (1990) found that the Baumberg marsh site in South San Francisco Bay subsided ~17 cm between 1930 and 1988 as a consequence of subsurface water extraction.

Along the northern and northwestern Gulf of Mexico, oil and gas extraction has caused measurable subsidence of coastal wetlands. For instance, oil and gas withdrawal in Louisiana has resulted in 2 cm of local coastal subsidence (MARTIN and SERDENGECTI, 1984; SUHAYDA, 1987; TURNER, 1990). In one area, the land surface has subsided by ~80 cm (SUHAYDA, 1987). According to BOESCH *et al.* (1994, p. 39), however, "only ~1% of the coast (~400 km^2) has a subsidence potential associated with oil and gas production greater than 10 cm over and above the natural subsidence caused by lithospheric downwarping and consolidation of sediments."

Wetland losses along the southeast Texas Gulf Coast have

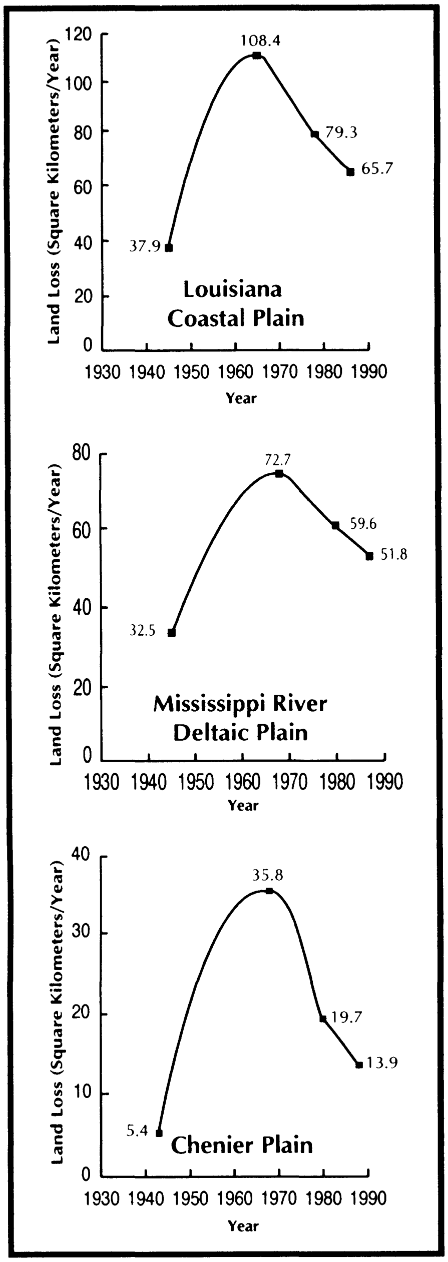

Figure 7. Coastal wetland loss from the 1940s to the 1980s in the Louisiana Coastal Plain, Mississippi River Deltaic Plain, and the Chenier Plain. From BRITSCH and DUNBAR (1993) and BOESCH *et al.* (1994).

been coupled to the development of surface faults associated with water, oil, and gas withdrawal. WHITE and MORTON (1997) specified that ~5,000 ha of marsh habitat have been lost since the 1930s and 1950s due to faulting activated in large part by hydrocarbon extraction (Figures 8a, b). In the Galveston Bay estuarine system, submergence has affected nearly 11,000 ha of wetlands habitat, including extensive areas of tidal salt marsh vegetation (WHITE and TREMBLAY, 1995). Inundation has converted much of the wetlands habitat to open water and shallow subaqueous flats, with shorelines retreating relatively rapidly landward. Groundwater withdrawal and the more localized effect of hydrocarbon production have significantly increased the rate of relative sea-level rise such that it greatly exceeds the rate of wetland vertical accretion in the area. Since 1906, groundwater withdrawal has caused up to 3 m of land surface subsidence in the Houston-Galveston area, encompassing an area of more than 9.4×10^5 ha (GABRYSCH and COPLIN, 1990). The rate of subsidence has ranged from 10 mm/yr to more than 60 mm/yr. A large subsidence bowl has formed as a result of long-term groundwater pumpage in this area, with ~10,700 ha of marsh converted to open water and flats from the 1950s to 1989 (WHITE *et al.*, 1993). However, the human-induced subsidence has declined considerably since the late 1970s in response to curtailment of groundwater withdrawal (WHITE and TREMBLAY, 1995).

Global Impacts

Relative sea-level change is influenced by an array of anthropogenic factors many of which are local or regional in nature and affect hydrology, soils, plant communities, sediment inputs, accretion rates, subsidence, and other components of coastal wetlands. Less conspicuous are human activities that potentially impact global sea level and nationwide coastal wetland systems. Most conspicuous in this regard are possible changes in world climate linked to global warming due to the increased burden of anthropogenic greenhouse gases and tropospheric sulfate aerosols in the atmosphere (PINET, 2000). Much of this burden derives from the combustion of fossil fuels (coal, oil, and gas).

Significant increases in several greenhouse gas concentrations have been documented over the industrial period. For example, CO_2 concentrations in the atmosphere increased from ~280 ppm in the preindustrial era to ~364 ppm in 1997; CH_4 concentrations increased from ~700 ppb in preindustrial times to ~721 ppb in 1994; and N_2O concentrations increased from ~275 ppb in preindustrial times to ~312 ppb in 1994. In addition, chlorofluorocarbons (CFC-11, CCL_3F; CFC-12, CCL_2F_2), manmade compounds which also constitute major greenhouse gas constituents, were not appreciably present in the atmosphere prior to 1950 (LEDLEY *et al.*, 1999).

Some global climate models predict a significant rise in eustatic sea level attributable to emissions of carbon dioxide in the atmosphere and the occurrence of the anthropogenic "greenhouse" warming (REVELLE, 1983; TITUS, 1986). Eustatic sea level is expected to rise another 20 to 115 cm during the 21st Century as a result of glacial melting from atmospheric warming; ocean water expansion is included in the range provided (WOODROFFE, 1993). CRAFT *et al.* (1993) suggest that eustatic sea level could rise up to 250 cm by the year 2075, which would drown great expanses of coastal habitats worldwide.

During the past two to four decades, Arctic ice cover has decreased by ~40% in thickness. Analyzing microwave satellite remote sensing data, JOHANNESSEN *et al.* (1999) ascertained that the areal extent of Arctic sea ice cover declined ~3% per decade between 1978 and 1998. Perennial sea ice

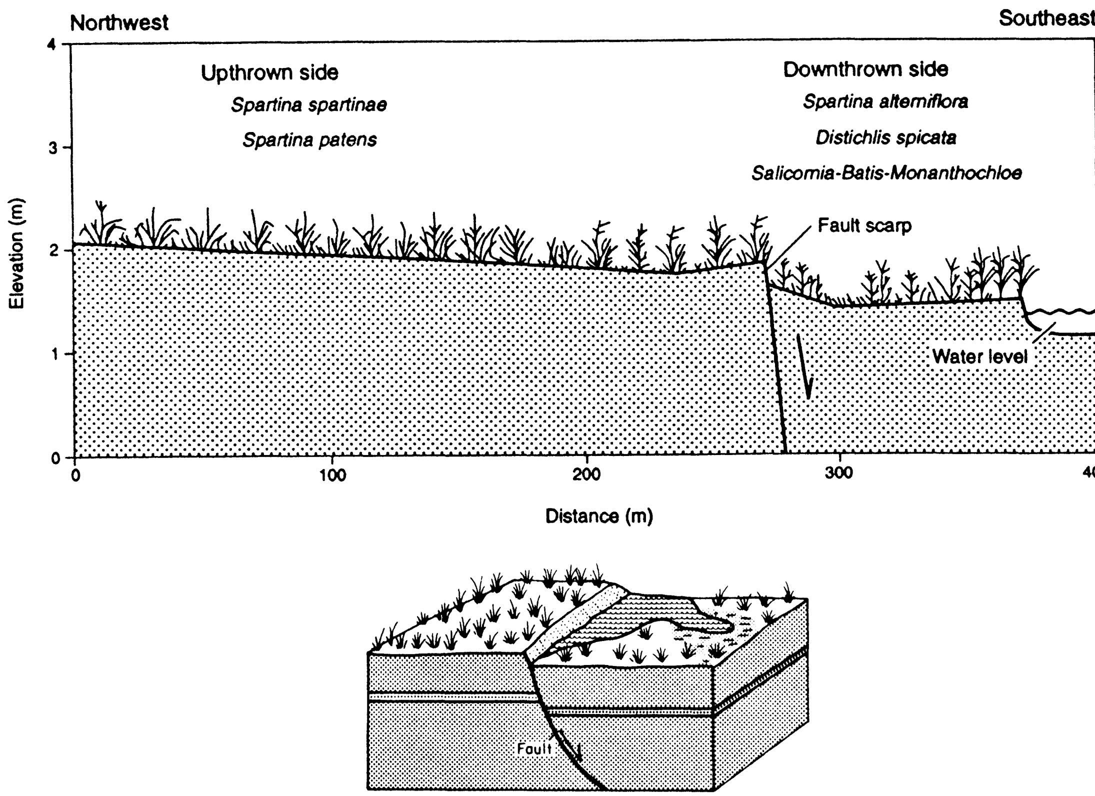

Figure 8. (a) Topographic profile across an active fault on the upper Texas Coast exhibiting relative elevations and plant communities on each side of the fault. From WHITE and PAINE (1992) and WHITE and MORTON (1997). (b) Block diagram of changes in wetlands that may occur along an active surface fault on the upper Texas Coast. From WHITE *et al.* (1993) and WHITE and TREMBLAY (1995).

declined by 0.031×10^6 km²/yr between these years, amounting to a drop of ~7% per decade. This reduction in multiyear ice area indicates that substantial rather than only peripheral changes have taken place in Arctic sea ice cover in recent years (JOHANNESSEN *et al.*, 1999; LEVI, 2000). Diminishing Arctic ice mass appears to represent a major climate signal, closely coupled to elevated emissions of greenhouse gases in the atmosphere. Climate models recently applied to trends of northern hemisphere sea ice data project a continued decrease in sea ice thickness and extent throughout the 21st Century (VINNIKOV *et al.*, 1999).

COASTAL COMPARISONS

Atlantic Coast

RAMPINO and SANDERS (1981) stated that the submergence rate of coastal salt marshes in the northeastern U.S. (New Jersey to Maine) depends on: (1) the regional submergence rate which is coupled to the collapse of a late Wisconsin ice-margin bulge; and (2) eustatic sea-level rise. They inferred that periods of tidal salt marsh growth correspond to times of reduced rate of submergence. According to the authors, the decreased rate of submergence in this region during the past few thousand years is probably mainly due to a decline in the eustatic component of sea-level rise.

Data from STEVENSON *et al.* (1986) on salt marsh systems from Delaware to Massachusetts reveal mean accretion rates of 4.3 to 5.5 mm/yr and relative rates of sea-level rise of 0.9 to 2.0 mm/yr. Relative sea level has increased in eastern Maine during the past century, rising at rates of 2.1 mm/yr in Portland and 2.5 mm/yr in Eastport between 1930 and 1990, compared to long-term mid- and late Holocene rates of ~1.0 to 1.4 mm/yr (GEHRELS, 1994). GEHRELS (1994) concludes that the long-term rates of relative sea-level rise in eastern Maine are close to those of isostatically and tectonically stable coastlines and that coastal subsidence does not appear to have played a significant role in the relative sea-level history of this region during the past several thousand years. Tidal salt marshes in eastern Maine have been able to keep pace with the relatively low rates of sea-level rise, a situation much different than that observed in parts of Chesapeake Bay and the Louisiana coastal zone. ANDERSON *et al.* (1992) demonstrated that human activities (*i.e.,* deforestation and agriculture) have contributed to increased land erosion and inorganic sediment supply to sheltered coves in eastern Maine in the past, thereby favoring marsh accretion and development. However, other anthropogenic factors (ditching and impoundment dikes) may be degrading some salt marsh systems (*e.g.,* Wequetequock-Pawcatuck tidal marshes in Connecticut) in the northeast region by reducing sediment supply and vertical accretion of the marsh surface (WARREN and NIERING, 1993).

PHILLIPS (1986) speculates that the New Jersey shore of Delaware Bay may not be representative of other Atlantic coastal plain estuaries. In this area, tidal salt marshes are experiencing a rapid areal loss, resulting from accelerated submergence of the Delaware Bay shoreline. Sea-level rise, subsidence, and dissection are hastening shoreline erosion. Mosquito ditching and salt hay farming have accelerated subsidence. Shoreline recession rates in this area are extreme, averaging 3.21 m/yr and causing a substantial loss of wetland habitat.

Shoreline erosion in Chesapeake Bay is also elevated, exceeding 3.0 m/yr in some regions (WARD *et al.*, 1988). The relative rate of sea-level rise averages ~3.0 mm/yr in the mid-bay region (STEVENSON *et al.*, 1986), with subsidence (1.6 to 2.0 mm/yr) accounting for most of the water level change (KEARNEY and STEVENSON, 1991). The high subsidence rates, which have eliminated extensive areas of marsh habitat on flat terraces (KEARNEY *et al.*, 1988), are largely attributed to groundwater withdrawal (overpumping of surficial aquifers) and sediment loading. Groundwater withdrawal is responsible for substantial subsidence in the lower Chesapeake Bay (DAVIS, 1987). Sediment loading may be generating isostatic downwarping of the Chesapeake basin. Approximately 8×10^8 mt (metric tons) of sediment have accumulated in the Virginia and Maryland portions of Chesapeake Bay during the last century and in the bay's upper reaches. There is a long history of anthropogenically-induced soil erosion in the southern Piedmont which has delivered much sediment to rivers and estuaries of the region (TIMBLE, 1974).

Vertical accretion of the marshes has not kept pace with relative sea-level rise, leading to inundation and deterioration of the marsh surface. The loss of marsh habitat in the Chesapeake Bay region is especially notable at the Blackwater National Wildlife Refuge (STEVENSON *et al.*, 1985), and the Nanticoke River (KEARNEY *et al.*, 1988). Land loss of marsh-dominated islands has been substantial; it involves three major processes: (1) perimeter erosion (by wave action); (2) channel formation and enlargement; and (3) interior pond formation. One bay island near the mouth of the Nanticoke River (*i.e.,* Bloodsworth Island) lost 579 ha or 26% of its total wetland area from 1849 to 1992 due to these processes (DOWNS *et al.*, 1994).

Investigations of estuarine marshes on the Eastern Shore of Chesapeake Bay have delineated significant spatial and temporal variations in vertical accretion rates (KEARNEY *et al.*, 1994). Even in the same overall depositional environment, considerable spatial variability in vertical accretion is evident. Disparities in vertical accretion rates are generally observed between levee and channelside sites and interior marsh areas. The vertical accretion rates are typically higher at shoreline/channel sites, but may not be statistically significant. However, differing tidal regimes and the larger hydraulic framework may contribute to wide variations in sediment accumulation across the marsh surface. At Monie Bay on the Eastern Shore of the bay, the mean long-term (last two centuries) accretion rate has averaged ~3.0 mm/yr.

Shoreline erosion of tidal salt marshes has likewise been documented in Pamlico Sound, North Carolina, although it is less severe than in Delaware Bay. PHILLIPS (1986) found that the more protected salt marsh habitat in Pamlico Sound is eroding at a mean rate of 0.79 m/yr compared to a mean rate of 0.91 m/yr for shoreline marshes. He advises that shoreline erosion rates greater than ~0.3 m/yr will probably result in a net loss of marshes along the Atlantic and Gulf Coastal Plain. Thus, salt marsh erosion data for Pamlico Sound indicate ongoing shoreline recession that may lead to serious shrinkage of marsh habitat during the 21st Century.

As in the case of tidal salt marshes in Chesapeake Bay, those in North Carolina estuaries exhibit variable vertical accretion rates. For example, CRAFT *et al.* (1993) reported that at a regularly flooded estuarine marsh at Oregon Inlet along the central North Carolina coastline, accretion rates at streamside and in the backmarsh equal 2.7 mm/yr and 0.9 mm/yr, respectively. The marsh is maintaining its elevation relative to sea-level rise at the streamside areas but not in the backmarsh. At an irregularly flooded estuarine marsh at Jacobs Creek, CRAFT *et al.* (1993) recorded accretion rates of 2.4 to 3.4 mm/yr, indicating that the marsh elevation is keeping pace with the rate of sea-level rise (1.9 mm/yr) in this area.

In bar-built estuaries along the southeastern coast of North Carolina, lagoonal salt marshes are vertically accreting at a rate of ~1.2 mm/yr (HACKNEY and CLEARY, 1987). Because the average relative sea-level rise here is ~1.9 mm/yr, the salt marshes cannot maintain their elevation through autochthonous production alone. In the past, marsh aggradation has occurred when large quantities of sediment were made available to the wetland habitats through the tidal inlets. However, anthropogenic activities related to the removal of sand for beach renourishment projects on developed barrier islands, as well as the dredging of inlet channels to maintain navigable waterways, have greatly reduced the potential inorganic sediment supply to the marshes. HACKNEY and CLEARY (1987) estimate that if all the sediment dredged from New Topsail Inlet between 1967 and 1985 (776,899 m^3) and Masonboro Inlet between 1957 and 1985 (5,800,000 m^3) had instead been added to the estuarine area, marsh vertical accretion near Topsail Inlet and near Masonboro Inlet would be 5 mm/yr and 18.8 mm/yr, respectively, more than enough to maintain the marsh surface elevation relative to rising sea level.

In Georgia salt marshes, artificial impoundments have been responsible for sediment deposition rates as high as 1 cm/month (EDWARDS and FREY, 1977). Highest rates of deposition occur in streamside-levee marshes. Proceeding into back-levee areas, deposition rates are much lower, and less variable than those along streamside levee marshes, a similar condition observed in the back marshes of the Gulf Coast region and other salt marsh systems worldwide. This is due, in part, to the dynamics of flood-tide waters which overlie the marsh surface for periods of time that are too brief for large quantities of sediment to settle from suspension. Assessment of sedimentation rates on the marsh surface must consider vegetation cover, tidal regime, sediment supply, marsh confinement, and eustatic sea-level rise (LETZSCH and FREY, 1980).

Gulf Coast

The broad, gently-sloping plain of the Gulf Coast supports extensive salt marsh systems which have experienced substantial areal losses during the past century due to both natural and anthropogenic factors as noted previously. Coastal subsidence associated with sediment accumulation and compaction exacerbated by various human disturbances of wetlands habitat for purposes of flood protection, water supply, maritime commerce, energy production, and wildlife management has resulted in severe losses of salt marsh vegetation. Coastal erosion has created an additional long-term problem. Approximately 80% of the total coastal wetland loss in the U.S. has occurred in Louisiana alone, where ~41% of the nation's coastal wetlands exists (BOESCH *et al.*, 1994). The conversion of salt and brackish marshes to open water accounts for nearly 60% of the wetland losses along the coast of the northern Gulf of Mexico (DAHL *et al.*, 1991; WHITE and TREMBLAY, 1995). Substantial reductions in wetland area and a corresponding increase in open water habitat are projected for this region, if the relative rate of sea-level rise exceeds 10 mm/yr (ORSON *et al.*, 1985).

Shoreline retreat caused by long-term coastal erosion has also had an impact on wetlands deterioration along the Gulf Coast, albeit less than that of land subsidence. The most acute retreat is evident in Louisiana where erosion has accounted for a significant fraction of the 2,500 km^2 of coastal wetlands area lost during the past 45 years. An additional 1,500 km^2 of wetlands habitat will be lost in the next 50 years if current rates of erosion and subsidence continue. However, protective barrier islands offshore of estuaries and wetlands are eroding at rates up to ~20 m/yr, and some of them may be completely eroded away within the next several decades. The loss of the protective barriers will greatly accelerate the rate of coastal salt marsh losses (WILLIAMS *et al.*, 1997).

At the present time, the most rapidly eroding area in the Gulf is the open coast of the Mississippi Delta Plain. Here, the annual rate of sea-level rise amounts to ~10 mm/yr, and shoreline erosion exceeds 10 m/yr in some areas. Hence, severe problems have developed through most of its extent, with an annual loss of hundreds of hectares of marshlands and a marked retreat of the shoreline taking place especially on the Gulf side of the barrier islands.

Parts of the Chenier Coast of Texas are also eroding rapidly due to a combination of reduced sediment supply (in part caused by impoundments) and a relatively high rate of sea-level rise (~5–7 mm/yr) (MORTON, 1979; NRC, 1987). According to MORTON (1993), ~70% of the Texas Coast has experienced erosion rates up to 10 m/yr. Similar to shoreline impacts observed along parts of the northern Gulf of Mexico, this erosion may pose a threat to the barrier island systems which are protecting the mainland coastal areas.

The northeast Gulf Coast from the Mississippi Delta to the Apalachicola Delta in Florida shows a range of shoreline patterns. Wave energy is highest east of the Mississippi Delta and west of the Apalachicola Delta, and longshore sediment transport in the region is generally from west to east (STONE and STAPOR, 1996). In developed areas, shoreline retreat along the mainland coast has been substantial, contributing to wetland retreat (GMP, 1993). Coastal erosion is conspicuous in the western sector of the northeastern coastal zone (DAVIS, 1997).

The Gulf peninsula of Florida consists of a complex of some 30 barrier islands and inlets that extend roughly north-south for ~230 km as part of the west-central barrier chain. The northern coastal section is typified by open marsh coast, and the southernmost coast, by mangroves. Except for some local erosion problems within the barrier system of the central coastal section, most of the Gulf peninsula coast of Florida remains relatively stable (DAVIS, 1997). Shoreline retreat is slight, and wetlands habitat, less impacted by human activities than in other regions of the Gulf Coast.

Although a general consensus exists regarding the rapid deterioration of coastal wetlands in the Gulf of Mexico, it is unclear whether natural factors or human activities are largely responsible for most of the habitat destruction. Wetland losses here would likely occur even without human intervention because of natural subsidence caused by the compaction of deltaic sediments and downwarping of the older Pleistocene surface, and diminishing sediment accretion (MITSCH and GOSSELINK, 1993). Nevertheless, various human activities have accelerated the destruction of coastal salt marshes in the Gulf, such as canal construction, failed impoundments, land reclamation, dredging, and fluid withdrawal. The impact of oil, gas, and groundwater withdrawal on wetlands appears to be largely localized to areas near shallow oil and gas fields or large cities (BOESCH *et al.*, 1994). Improved wetland management and restoration programs, together with regulatory protection measures, have reduced the loss of coastal salt marsh habitat in recent years, but a much more comprehensive management and restoration effort must be mounted to counter the devastating losses of wetlands habitat during the 20th Century.

Pacific Coast

Salt marsh systems along the Pacific Coast differ substantially from those along the Atlantic and Gulf Coasts due to physiography and tectonic activity. For example, salt marshes surrounding the steeply sloping bays of New England are less extensive than those occupying the gently-sloping plains from the Mid-Atlantic region to the Gulf Coast, and the tectonically active Pacific Coast is rather depauperate in coastal salt marsh vegetation. When present, Pacific Coast salt marshes are often confined to relatively narrow fringes of lagoons and sheltered bays. Although they typically have a rather restricted distribution, Pacific Coast salt marshes characteristically exhibit greater floral diversity than those along other U.S. coastal regions, and their zonation and succession patterns appear to be more complex (MACDONALD, 1977). The plant associations are distinctly different in New England marshes, Atlantic and Gulf coastal plain marshes, and Pacific marshes (FREY and BASAN, 1985; MITSCH and GOSSELINK, 1993).

Along the Pacific Coast, subsidence exerts less of an influence than tectonic activity on the structure of salt marsh systems. Because of the lack of general subsidence along the Pacific Coast equal in magnitude to that encountered along the Gulf Coast, Pacific Coast salt marshes have largely been able to keep pace with the small increase in eustatic sea-level rise to maintain their elevation and viability (PATRICK and DELAUNE, 1990). Relatively few investigations of marsh accretion have been conducted on the Pacific Coast. Where vertical accretion has been studied (*e.g.,* northwest Pacific marshes), marsh surfaces appear to be effectively maintaining their elevation relative to rising sea level. For example, in Oregon and Washington, THOM (1992) reported a mean accretion rate of coastal salt marshes amounting to 3.6 mm/yr, far exceeding the mean rate of global sea-level rise (1.3 mm/yr). Similarly, PATRICK and DELAUNE (1990) registered mean accretion rates of 5 mm/yr, 8 mm/yr, and 42 mm/yr at Bird Island, Baumberg, and Alviso marsh sites, respectively, in South San Francisco Bay. As in the case of salt marshes in Oregon and Washington, those bordering South San Francisco Bay are accreting sufficient sediment to maintain the marsh surface within an appropriate tidal range to ensure the viability and persistence of the habitat.

Human activities also impact Pacific Coast salt marshes. As shown by THOM (1992), for instance, road beds and agriculture dikes constructed throughout salt marsh systems in the northwest Pacific, adversely affect the marsh surface morphology, as well as the biomass and composition of plant assemblages. In the San Francisco Bay region, groundwater withdrawal has increased subsidence by ~1 m between 1960 and 1990. However, mineral sedimentation and peat formation have been adequate to compensate for the high rates of subsidence and the low rates of sea-level rise to maintain the elevation of the marsh surface above mean high water (PATRICK and DELAUNE, 1990). For healthy salt marsh vegetation to be maintained in the face of ongoing human impacts, relatively high rates of sedimentation must continue in these dynamic coastal systems.

SUMMARY AND CONCLUSIONS

More than half of the original salt marsh habitat in the U.S. has been lost, due in large part to development and human activities in coastal regions. Approximately 125 million people currently reside in coastal counties nationwide, placing considerable pressure on these vitally important systems. Poorly planned urban and industrial expansion, the advent of suburban sprawl, and uncontrolled growth of many coastal settlements greatly accelerated the destruction of salt marshes during the 20th Century. Multiple uses and demands on coastal environments have stressed many salt marsh systems beyond their capacity to absorb human impacts, resulting in the loss and alteration of many hectares of valuable habitat.

There is a wide array of human activities that has directly affected tidal salt marshes. Included here are effects of agriculture (*e.g.,* salt hay farming), oil and gas production, waste disposal, transportation (airports, highways, and railroads), and other types of human intervention. Significant deterioration of salt marshes at the local scale has resulted from diking, ditching, canal construction, levee formation, infilling and drainage of wetlands area, dredging and dredge material disposal, as well as modified river catchments, and altered marsh hydrology. At the regional scale, coastal subsidence coupled to groundwater, oil, and gas withdrawal, together with vertical accretion deficits, has contributed to the submergence of extensive salt marsh habitats and their conversion to open water systems. This is particularly evident in the northern and northwestern Gulf of Mexico, and the lower Chesapeake Bay region. At the global scale, eustatic sea-level rise, likely coupled to global warming, also is constraining salt marsh habitats worldwide, albeit at a less rapid rate.

Glacial melting and thermal expansion of ocean waters have increased with escalating burdens of anthropogenic greenhouse gases and tropospheric sulfate aerosols in the atmosphere. Eustatic sea level is increasing at a rate of ~0.12 to 0.24 cm/yr, leading to a general retreat of ocean shorelines worldwide. The gradual inundation of ocean waters will play a greater role in salt marsh deterioration in many regions of the world during the 21st Century.

The inability of many salt marsh systems to maintain their elevation relative to sea level is a cause of increasing concern to marine and coastal scientists, land-use planners, decision makers, federal and state legislators, and interested members of the public. In some regions, the problem is mainly one of subsidence, often attributable to fluid withdrawal; in other regions, it is principally one of low vertical accretion ascribable to reduced riverine (mineral) sediment supply or diminished *in situ* organic soil formation. As a result, many coastal regions in the U.S. are now experiencing rapid relative sea-level rise and widespread wetland loss.

The causes and effects of relative sea-level change differ in coastal regions nationwide. Along the east coast of the U.S., for example, eustatic sea-level rise is generally the predominant factor in shoreline retreat, although subsidence coupled to groundwater withdrawal may be locally important. Along the Gulf Coast, global sea-level rise is of far less significance than subsidence in generating wetland loss. Here, subsidence precipitated by consolidation of deltaic sediments, crustal downwarping, and subsurface fluid withdrawal far outweighs other factors in controlling the survival/maintenance of salt marsh habitat. Along the west coast, tectonic activity triggered by lithospheric plate movements accounts for local changes in relative water levels that have caused a decrease in marsh surface area at some locations.

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